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Research article

Nutrient-derived environmental impacts in Chinese agriculture during 1978–2015

Huijun Wu [a], Shun Wang [a], Liangmin Gao [a], Ling Zhang [b], Zengwei Yuan [c, *], Tingyu Fan [a], Kaiping Wei [a], Li Huang [a]

[a] School of Earth and Environment, Anhui University of Science and Technology, Huainan 232001, PR China
[b] College of Economics and Management, Nanjing Forestry University, Nanjing 210037, PR China
[c] State Key Laboratory of Pollution Control and Resource Reuse, School of the Environment, Nanjing University, Nanjing 210023, PR China

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ABSTRACT

Nitrogen (N) and phosphorus (P) play a critical role in agricultural production and cause many environmental disturbances. By combing life cycle assessment (LCA) method with the mass balance principle of substance flow analysis (SFA), this study establishes a nutrient-derived environmental impact assessment (NEIA) model to analyze the environmental impacts caused by nutrient-containing substances of agricultural production in China during 1978–2015. The agricultural production system is composed of crop farming and livestock breeding, and the environmental impacts include energy consumption, global warming, acidification, and eutrophication. The results show all these environmental impacts had increased to $8.22*10^9$ GJ, $5.01*10^8$ t CO_2-eq, $2.41*10^7$ t SO_2-eq, and $7.18*10^7$ t PO_4^{3-}-eq, respectively. It is noted the energy consumption and the climate change caused by the crop farming were always higher than those from livestock breeding, which were average 60 and two times, respectively. While the acidification and the eutrophication were opposite after 1995 and 2000, even they were similar. This was mainly due to the high N application including synthetic N fertilizer (from $1.33*10^9$ GJ to $2.08*10^9$ GJ), applied manure (from $4.94*10^8$ GJ to $5.65*10^8$ GJ) and applied crop residue (from $2.94*10^8$ GJ to $5.30*10^9$ GJ), while the synthetic N fertilizer was controlled and the livestock expanded rapidly after 1995. Among the sub-categories, the three staple crops (rice, wheat, and maize) contributed greater environmental impacts, which were about two to 10 times as other crops and livestock, due to their high fertilizer uses, sown areas and harvests. While the oil crops and fruit consumed the least energies because of their much lower fertilizer-use intensities. Pig and poultry especially pig also caused obvious effects on environment (even 20 times as other livestock) because of their large quantities and excretions, which emitted much higher N_2O and P loss resulting in much higher climate change, acidification and eutrophication than other livestock. Then the study proposes the nutrient management in agricultural production by considering crop production, livestock breeding and dietary adjustment, so that some valuable experiences can be shared by the stakeholders in other Chinese regions.

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1. Introduction

As a result of ongoing population growth and diet transition to meat consumption, the demand for crop- and animal-derived food has greatly increased during the last few decades. To support food security, agricultural production develops rapidly, which has also caused severe environmental problems, restricting agricultural development and food security in turn (Chen et al., 2014; Godfray et al., 2010; Vermeulen et al., 2012). This challenge may grow in the future, as global food demand is likely to double by 2050 cased by the population growth (Chen et al., 2014). As the largest developing country, China's agricultural production is essential and has greatly increased to support its economic growth and meet with the food demand of its large population. The increase rate of chemical fertilizer application had achieved to six times of crops (NBSC, 2016). Also, animal production has raised rapidly, mainly due to the Chinese dietary preference and increased imports of feed (Ma et al., 2012). However, the rapid development of Chinese

* Corresponding author. Tel.: 86 025 89680532.
E-mail address: yuanzw@nju.edu.cn (Z. Yuan).

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agriculture also has brought many environmental pressures. For example, Chinese agriculture has consumed nearly 1/4 world's total agricultural energy, which correspondingly emitted annual GHG (greenhouse gas) emissions to over two times since 1978 (Fei and Lin, 2016; Luukkanen et al., 2015; Yu, 2016). The situation of growing contribution of agricultural sources to water quality degradation is now deteriorating even further (Chen et al., 2010). Thus, guaranteeing the increasing food needs while simultaneously reducing the environmental impact from agriculture is undoubtedly one of the greatest challenges of the century (Foley, 2011; Godfray et al., 2010; Makowski et al., 2014).

Nutrients such as nitrogen (N) and phosphorus (P) play a critical role in agricultural production and global food security (Erisman et al., 2008; Ma et al., 2012). They are contained in resources (eg. fertilizers, seed, and feed), products (eg. crops, meat, egg, and milk), and wastes (eg. crop residue, manure), flowing through the agricultural production including crop farming and livestock breeding (Chen et al., 2010; Fernandez-Mena et al., 2016). With the agricultural development, these nutrients flow quickly and massively. Large fractions of anthropogenically mobilized N losses through emissions of ammonia (NH_3), nitrous oxide (N_2O), and nitrogen oxide (NO_x). Also, large fractions of anthropogenically mobilized N and P flows into groundwater through runoff and leaching (Bouwman et al., 2013; Uwizeye et al., 2016). These have led to a series of environmental disturbances, including climate change, water eutrophication, and acidification (Fernandez-Mena et al., 2016; Sutton et al., 2013). For example, IPCC (2007) indicates agriculture currently accounts for approximately 14% of total global anthropogenic greenhouse gas emissions, and is responsible for about 58% of total anthropogenic emissions of N_2O. As the largest consumer of synthetic N in the world, Chinese agriculture has contributed 74% of the total N_2O emission in the country (NDRC, 2012). Fu et al. (2007) showed that more than 60% of the major lakes and reservoirs were eutrophic due to the high concentrations of total P and N in water. The 1st National census of pollution sources in China (MOEP and MOA, 2010) also identified the P loss from agricultural as the dominant contributor to the total P losses. It is therefore an urgent need to manage these nutrients to improve environmental performance in agriculture.

In fact, many studies have been published, focusing on the agricultural nutrient management. Earlier studies mainly evaluated nutrient inputs and outputs by using mass balance principle (Baker et al., 2001; Nilsson, 1995). They are believed to be appropriate for evaluating the nutrient flows and losses to determine the water eutrophication level. However, they treated systems as “black boxes”, which is difficult to identify sources of environmental impacts. Consequently, a systematic method named substance flow analysis (SFA) is proposed to improving nutrient management (Brunner, 2010). It is used to analyze flows and stocks of a single substance or of a coherent group of substances based on the mass balance principle (van der Voet, 2002). Concerning the importance of N and P in the social development, many SFA studies have examined nutrient flows at global (Canfield et al., 2010; Fowler et al., 2013; Chen and Graedel, 2016), national (Hamilton et al., 2017; Pearce and Chertow, 2017), regional (Asmala et al., 2011) and municipal (Thitanuwat et al., 2016) levels. China has also made great efforts to research nutrient flows. Liu (2004) and Chen and Chen, 2008 early analyzed P flows and P losses to water environment, providing the research basis of other related studies in China (Cui et al., 2013a; Wu et al., 2015; Gu et al., 2015; Liu et al., 2016). Apart from nutrient flow evaluation, Ma et al. (2013) and Wu et al. (2015) proposed improving the nutrient use efficiency in the food system of China. There are also many related studies especially in agricultural production and consumption in different regions, including watersheds (Asmala et al., 2011; Jiang and Yuan, 2015), provinces (Wu et al., 2014; Zhang et al., 2016a), and cities (Cui et al., 2015; Lin et al., 2016; Wu et al., 2012a). These studies examined nutrient flows systematically from the life-cycle aspect. However, emphasizing nutrient flow could only focus more on nutrient loss to surface water, while seldom consider other environmental impacts (e.g., energy consumption, climate change, land use, etc.). Thus, the SFA method cannot be solely implied to analyze the more varied environmental impacts and generate comprehensive recommendations.

Life cycle assessment (LCA) is a methodology to assess all the environmental impacts associated with a product, process or activity by identifying, quantifying and evaluating all the resources consumed, and all emissions and wastes released into the environment (Guinée, 2002). It is shown to be a systematic method analyzing the environmental impacts more comprehensively and more objectively (Wu et al., 2017). LCA method is mostly used in the industrial sector, such as energy (Garrido et al., 2017), building (Vitale et al., 2017; Wu et al., 2012b), and waste disposal (Nabavi-Pelesaraei et al., 2017). In recent years, LCA research began to pay more attentions on agriculture, and its derived environmental impacts (eg. climate change, acidification, eutrophication, etc.) (Foteinis and Chatzisymeon, 2016; Wu et al., 2016). Among these studies, the nutrients are also the research focus because of their key importance in agriculture (Ridoutt et al., 2013; Linderholm et al., 2012). While the nutrient-LCA studies are often conducted on some certain materials, such as fertilizers (Zhang et al., 2013; Hasler et al., 2015), crops (Zhang et al., 2016b), and wastes (ten Hoeve et al., 2014), or some certain sub-sectors, such as breeding (de Vries et al., 2015). They seldom consider the whole agricultural system composed of different agricultural activities (eg. cropping, breeding) and materials (eg. fertilizer, feed, nutrient leaching), which would provide more systematic analysis and comprehensive strategies. Moreover, the nutrient-based LCA is a more micro-method and some data are hardly collected or calculated directly, which would affect evaluating the environmental impacts. Consequently, combining the LCA method with the mass balance principle of SFA would be a more efficient method to analyze the nutrient-derived environmental impacts in the agriculture.

Here, this study analyzes the historical changes of environmental impacts derived by N and P in Chinese agricultural production system. The analysis consists of four parts: (a) establishing a model characterizing the nutrient-derived environmental impacts in agricultural production; (b) analyzing the environmental impacts caused by nutrients in the agricultural production of China during 1978–2015; (c) comparing the environmental impacts among the different crops and livestock; (d) proposing the nutrient management to mitigate the hotspots of these environmental impacts.

2. Methods

2.1. Logical framework of modeling

In this study, the environmental impacts in the agricultural production are quantified by using the nutrient-derived environmental impact assessment (NEIA) model. The model is established by combing LCA method with the mass balance principle of SFA. A schematic representation of the NEIA model is given in Fig. 1. Basing on the studies reviewed above, this study analyzes four environmental impacts including energy consumption, global warming, acidification, and eutrophication, which are mainly caused by the nutrient inputs and losses. Moreover, the agricultural production consists of crop farming and livestock breeding. Considering the great proportions of farming and breeding, nine crops (rice, wheat,

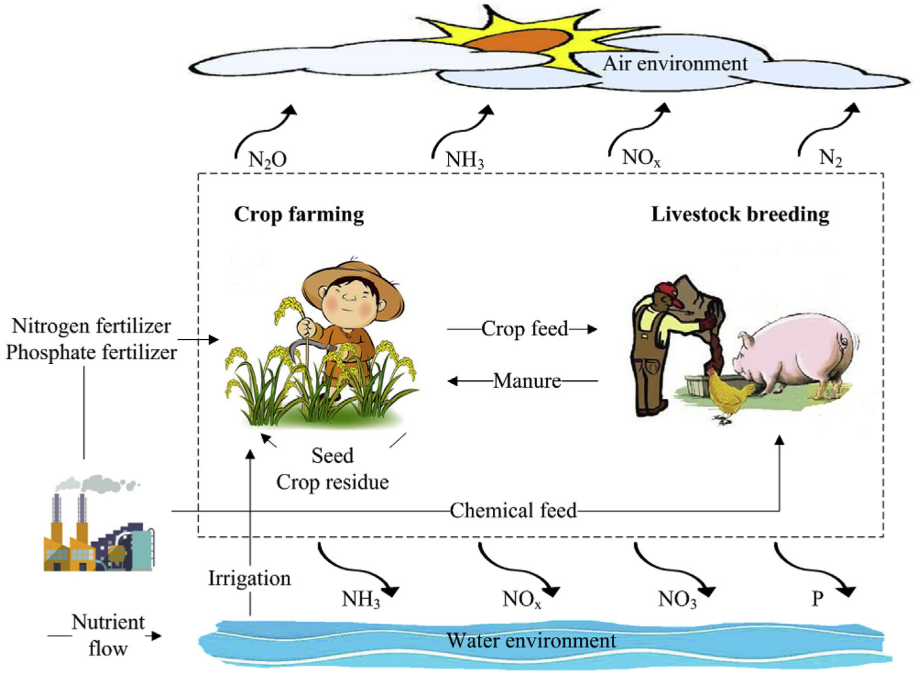

Fig. 1. The logical framework of the model of the nutrient-derived environmental impacts in the agricultural production.

maize, bean, cotton, peanut, rapeseed, vegetable, and fruit) and four livestock (pig, cattle, sheep, poultry) are selected. It should note the study only consider the energy consumption and emissions originated from N and P flows (eg. NH_3, N_2O, NO_x, P loss, etc.), other substances such as electricity consumption, methane (CH_4), and carbon dioxide (CO_2) are not included.

(a) Crop farming. This subsector involves the planting of crops. The nutrient inputs mainly refer to the consumed energy, including nitrogen fertilizer, phosphate fertilizer, irrigation, seed, manure, and crop residue. These inputs generate the next node with harvested grains and crop residue (mainly straw). Other inputs such as atmospheric deposition and biological N_2 fixation are not considered because of their low proportions. It should also note the manure and the crop residue refer to the agricultural resources applied to the field, which are parts of the total manure and crop residue. The nutrient losses are associated with emission of NH_3, N_2O, NO_x, and dinitrogen (N_2) to the atmosphere, and the NO_3 leaching to water (Brentrup et al., 2004b; Chen et al., 2010; de Vries et al., 2015; Ma et al., 2010; Ma et al., 2014a). In addition, P losses from cropland occur mainly as a result of erosion and runoff (Liu et al., 2008). There are have been many studies researching the impact of P on the water environment (Chen and Chen, 2008; Chen et al., 2008; Jiang and Yuan, 2015; Liu et al., 2008; Nilsson, 1995). Basing on these research, the study assumes that P losses are discharged into surrounding water (de Vries et al., 2015).

(b) Livestock breeding. With the feed and livestock products (meat, eggs, and milk) being consumed and produced, the manure is discharged, which emits nutrients to atmosphere and groundwater. These nutrient losses include N_2O, NH_3, NO_x, N_2, NO_3 leaching and P loss to groundwater (Bouwman et al., 2013; Chen et al., 2010; Ma et al., 2010; Uwizeye et al., 2016).

2.2. Quantification methods

As the purpose of the study is to analyze the potential environmental impacts derived by nutrients in the agricultural production, the model focuses on those impact categories through the whole life cycle of agricultural production. Each impact category (eg. acidification) could be given by multiplying their corresponding environmental indicators (eg. NH_3 and NO_x) with the characterization factors, which represent the potential of a single emission or resource consumption contributing to the respective impact category (ISO, 2006). Table 1 gives a list of the characterization factors for the aggregation of single environmental indicators for each impact category. It is noted the energy consumption is evaluated by multiplying the nutrient-containing agricultural resources with their corresponding energy equivalents, as illustrated in detail in the supplementary material. The quantification method of every impact category in each process of agricultural production is as following. The detailed calculation method is listed in Table S1 of the supplementary material.

It should be highlighted the study period ranges from 1978 to

Table 1
Characterization factors for the aggregation of single environmental indicators for each impact category.

Impact category	unit	Contributing indicator	Characterization factor	References
Global warming potential	kg CO_2-eq	N_2O	298	(IPCC, 2007)
Acidification potential	kg SO_2-eq	NH_3	1.88	(Brentrup et al., 2004a; Huijbregts, 2001)
		NO_x	0.70	
Eutrophication potential	kg PO_4^{3-}-eq	NH_3	0.33	(Brentrup et al., 2004a; Huijbregts, 2001)
		NO_x	0.13	
		NO_3	0.42	
		TP	3.06	

2015. The reason of choosing 1978 as the start year is the reform and opening-up in China was launched in 1978, when China has experienced too much in policy, economy and society. Meanwhile, in order to simplify the calculation, other years 1985, 1990, 1995, 2000, 2005, 2010, and 2015 were selected to show the trend of these nutrient-derived environmental impacts.

2.2.1. Energy consumption potential

This issue relates to the consumption of nutrient-containing energy. The consumption of different energy is aggregated to one summarizing indicator for energy consumption with energy equivalent. In crop farming, the consumed energy includes nitrogen fertilizer, phosphate fertilizer, irrigation, seed, manure, and crop residues. While in livestock breeding, the energy only refers to the feed consumed by pig, cattle, sheep, and poultry. The related calculation methods are illustrated by the following equations.

$$E_k = E_k^n + E_k^p + E_k^i + E_k^s + E_k^m + E_k^r (k = 1, 2, 3, \ldots, 9) \tag{1}$$

where $E_k(k = 1, 2, 3, \ldots, 9)$ is the energy consumption of every crop (rice, wheat, maize, bean, cotton, peanut, rapeseed, vegetable, and fruit) in crop farming. $E_k^n(i = 1, 2, 3, \ldots, 9)$, $E_k^p(k = 1, 2, 3, \ldots, 9)$, $E_k^i(k = 1, 2, 3, \ldots, 9)$, $E_k^s(k = 1, 2, 3, \ldots, 9)$, $E_k^m(k = 1, 2, 3, \ldots, 9)$, $E_k^r(k = 1, 2, 3, \ldots, 7)$ illustrate the energies embodied in nitrogen fertilizer, phosphate fertilizer, irrigation, seed, manure, and crop residues for the crop respectively.

$$E_i^{'} = E_i^f (i = 1, 2, 3, 4) \tag{2}$$

where $E_i^{'}(i = 1, 2, 3, 4)$ refers to the energy consumed by livestock (pig, cattle, sheep, and poultry) in livestock breeding.$E_i^f(i = 1, 2, 3, 4)$ is the energy embodied in the feed consumed by the livestock.

2.2.2. Global warming potential

The main anthropogenic N loss to the climate change is N_2O from farming and breeding, which can be calculated according to equations (3) and (4). The detailed calculation of N_2O is illustrated in Table S1, according to IPCC (2007). It could be found this greenhouse gas from farming is mainly determined by N application consisting of chemical fertilizer, manure, and crop residue. While the N_2O in breeding is emitted from the manure discharged by the livestock.

$$C_k = C_k^{N_2O} c^{N_2O} (k = 1, 2, 3, \ldots, 9) \tag{3}$$

where $C_k(k = 1, 2, 3, \ldots, 9)$ is the global warming potential caused by every crop (rice, wheat, maize, bean, cotton, peanut, rapeseed, vegetable, and fruit). $C_k^{N_2O}(k = 1, 2, 3, \ldots, 9)$ is N_2O emission of every crop from farming. c^{N_2O} represents the characterization factor of N_2O illustrated in Table 1.

$$C_i' = C_i^{N_2O\prime} c^{N_2O} (i = 1, 2, 3, 4) \tag{4}$$

where $C_i^{'}(i = 1, 2, 3, 4)$ illustrates the global warming potential contributed by the manure discharged from every livestock (pig, cattle, sheep, and poultry). $C_i^{N_2O'}(i = 1, 2, 3, 4)$ refers to N_2O emission from the manure of every livestock.

2.2.3. Acidification potential

As illustrated in equations (5) and (6), the acidification is mainly caused by air emissions of NH_3 and NO_x. In crop farming, these acid gases are mainly emitted from N application including chemical fertilizer, manure, and crop residue. While in livestock breeding, these gases primarily originates from the discharged manure.

$$A_k = A_k^{NH_3} a^{NH_3} + A_k^{NO_x} a^{NH_x} (k = 1, 2, 3, \ldots, 9) \tag{5}$$

where $A_k(k = 1, 2, 3, \ldots, 9)$ denotes the acidification potential contributed by every crop (rice, wheat, maize, bean, cotton, peanut, rapeseed, vegetable, and fruit) in farming. $A_k^{NH_3}(k = 1, 2, 3, \ldots, 9)$ and $A_k^{NO_x}(k = 1, 2, 3, \ldots, 9)$ indicate NH_3 and NO_x emissions from every crop. a^{NH_3} and a^{NH_x} are the characterization factors of NH_3 and NO_x seen in Table 1.

$$A_i' = A_i^{NH_3\prime} a^{NH_3} + A_i^{NH_x\prime} a^{NH_x} (i = 1, 2, 3, 4) \tag{6}$$

where $A_i'(i = 1, 2, 3, 4)$ refers to the acidification potential of every livestock (pig, cattle, sheep, and poultry) in breeding. $A_i^{NH_3'}(i = 1, 2, 3, 4)$ and $A_i^{No_x'}(i = 1, 2, 3, 4)$ are NH_3 and NO_x emissions from every livestock.

2.2.4. Eutrophication potential

The anthropogenic N and P emissions to surface water causing eutrophication are composed of two parts from farming and breeding. One is the deposition of NH_3 and NO_x on surface water. The other is the P loss and NO_3 leaching (Klepper et al., 1995). It should be noted the NH_3 and NO_x could not only cause the acidification, but also cause the eutrophication. Thus, the amounts of these gases are the same in calculating acidification potential and eutrophication potential. The calculations of eutrophication potential of farming and breeding are illustrated by equations (7) and (8) respectively.

$$U_k = A_k^{NH_3} u^{NH_3} + A_k^{NO_x} u^{NO_x} + U_k^{NO_3} u^{NO_3} + U_k^p u^p (k = 1, 2, 3, \ldots, 9) \tag{7}$$

where $U_k(k = 1, 2, 3, \ldots, 9)$ is the eutrophication potential contributed by the farming of every crop (rice, wheat, maize, bean, cotton, peanut, rapeseed, vegetable, and fruit). $U_k^{NO_3}(k = 1, 2, 3, \ldots, 9)$ and $U_k^p(k = 1, 2, 3, \ldots, 9)$ are NO_3 leaching and P loss from the farming of every crop. u^{NH_3}, u^{NO_x}, u^{NO_3}, and u^p are the characterization factors of NH_3, NO_x, NO_3, and TP shown in Table 1.

$$U_i' = A_i^{NH_3\prime} u^{NH_3} + A_i^{NO_x\prime} u^{NO_x} + U_i^{NO_3\prime} u^{NO_3} + U_i^{p\prime} u^p (i = 1, 2, 3, 4) \tag{8}$$

where $U_i'(i = 1, 2, 3, 4)$ is the eutrophication potential generated from the breeding of every livestock (pig, cattle, sheep, and poultry). $U_i^{NO_3\prime}(i = 1, 2, 3, 4)$ and $U_i^{p\prime}(i = 1, 2, 3, 4)$ represent NO_3 leaching and P loss from the breeding of every livestock.

2.3. Data sources

Basic data were derived from governmental statistical yearbooks, literature, and molecular calculation. Further information on the data sources relating with the NEIA model of the agriculture is presented in Table S2 of the supplementary material. Other basic data demonstrating the agricultural characteristics of the agricultural production in China during 1978–2015 is shown in Table S3.

The statistical data were gained from governmental statistical yearbooks (NBSC, 2016; PDNDRC, 2001, 2003, 2006, 2011, 2016), covering sown areas of crops, application densities of chemical fertilizers, irrigation areas, amounts of seeds per sown area,

harvests of crops, and amounts of livestock. Data related with nutrient-containing rates of substances (eg. chemical fertilizers, crops, and crop residues), energy equivalents of agricultural inputs (eg. chemical fertilizers, seed, irrigation, and feed), manure discharged per livestock, proportions of manure and crop residue applied to field, and emission factors of some gases were collected from published literature. Other data, such as conversion ratios of N to N-containing gases (eg. N_2O, NH_3, NO_x, and NO_3) and P-containing rate of P_2O_5 were calculated by molecular formula directly.

3. Results and discussion

In fact, there are many parameters being used for the unit cultivated land area and the unit livestock, such as the application densities of chemical fertilizers, energy equivalent of irrigation, nutrient excretion per livestock, and energy equivalent of feed per livestock, etc. Thus this study analyses the total value of every environmental impact, not considering the impacts per cultivated land area or livestock.

3.1. The total environmental impacts from agriculture

Fig. 2 displays the four environmental impacts of Chinese agricultural production during 1978–2015. It also illustrates the contributions to these impacts from crop farming and livestock breeding. According to these results, all the environmental impacts increased including energy consumption, which increased from $2.45*10^9$ GJ to $8.22*10^9$ GJ (Fig. 2(a)). It is noted the energy consumption was mainly determined by that from crop farming, which was much higher than that from livestock breeding. Fig. 3 shows the distributions of the nutrient-containing energy inputs to the total energy consumption. Among these energy inputs, nitrogen fertilizer, applied manure, and applied crop residue are the three largest energy inputs. Because of the increased food demand and the raised rural residents' income, the application of nitrogen fertilizer grew quickly. China is now the largest consumer of nitrogen fertilizer in the word, accounting for about 30% of the world's total use (Zhang et al., 2013). This has also resulted in the increasing of N-related energy consumption, which is almost higher than other energy consumptions. However, contrast to the mechanized and integrated nutrient management practices widely adopted in developed countries, Chinese farmers often apply fertilizer manually to the small plots, resulting in gross over-application (Ju et al., 2009). In the1990s, the government and the scientists correspondingly began to raise concerns and make efforts over the overuse and environmental impacts, and since then the fertilizer application has decreased and grew slowly. In addition, the high use of fertilizer also accelerated the crop harvest. Guo et al. (2010) showed a three-fold increase in N fertilizer application in

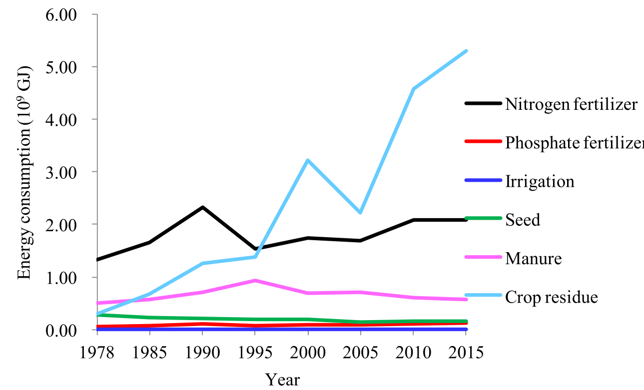

Fig. 3. Energy consumption of crop farming in China during 1978–2015.

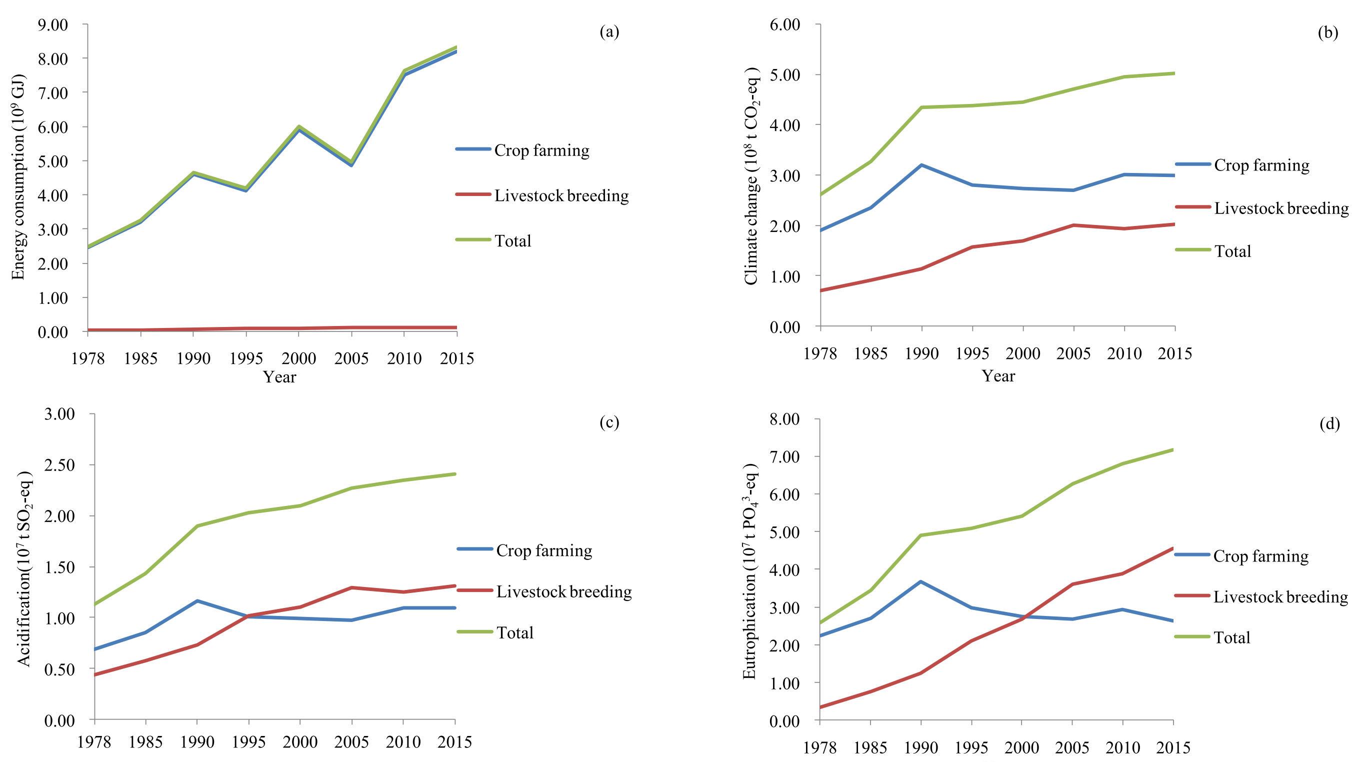

Fig. 2. Nutrient-derived environmental impacts in Chinese agriculture during 1978–2015.

agriculture has contributed to a near 70% increase in crop harvest in China since 1980. The energy embodied in applied crop residue increasing obviously from 2.94*10^8 GJ to 5.3*10^9 GJ, and even exceeding the nitrogen fertilizer and applied manure after 1995. These mainly due to two reasons. Firstly, the harvests of all the crops especially vegetable and fruit grew rapidly during these years, which also resulted in the increase of the crop residues. Then, the proportion of crop residue applied to field also increased quickly in these years, according to some studies shown in Table S2. Moreover, China has the world's largest livestock population that produces large amount of manure annually (Devendra, 2007). In these years, the rural residents also occupied over 50% of Chinese population, which is the largest in the world (NBSC, 2016). While the infrastructures required to dispose of waste in rural areas and some breeding enterprises were backward in the early years (Wu et al., 2015). These contributed the applied manure discharged by livestock and rural residents could not be ignored, the energy embodied in which was the third largest after nitrogen fertilizer and applied crop residue. While this energy decreased from 9.39*10^8 GJ after 1995. The main reason maybe the majority of the excrement was stored and dumped because of the increasing use of flush toilets and the popular application of inorganic fertilizers (Chen and Tang, 1999).

The climate change potential of the agricultural production is shown in Fig. 2(b), which tripled to 5.01*10^8 t CO_2-eq between 1978 and 2015. Compared with the livestock breeding, the crop farming contributed more to the total climate change potential, accounting for 60%–73%. This was mainly due to the N application especially the nitrogen fertilizer and manure (Fig. 4). As discussed above, much nitrogen fertilizer and manure applied in farming. And there are many studies (Kahrl et al., 2010; Ma et al., 2014a; Zhang et al., 2013) pointed out the application of N fertilizers in China is an important contributor to GHG emissions, especially N_2O. It should note the climate change potential decreased obviously after 1990, and increased slowly from 2005. This trend was the same as that of application of nitrogen fertilizer, the reason of which has also been discussed. Concerning the trend of livestock breeding, the climate change also increased rapidly, especially after 1995. This is mainly due to the following reasons. In fact, with the progressive increase in demand for meat, the livestock is rapidly expanding. Especially the pig grew sharply after 1990s, while the pig often contributes the most in N_2O emission from manure management (Zhou et al., 2007).

As noted, acidification is mainly contributed by NH_3 and NO_x. In crop farming, NH_3 is 10.0% of the total N application (Wang et al., 2002, 2007; Zhu and Chen, 2002), and NO_x is 10% of N_2O (Brentrup et al., 2004b), which is also determined by the N application (IPCC, 2007). While in livestock breeding, both NH_3 and NO_x have positive correlation with the manure (Klimont and Brink, 2004; Ma et al., 2012; Skiba et al., 1997). Hence, the change of total acidification potential of agricultural production was the same as the total climate change, growing from 1.13*10^7 t SO_2-eq to 2.41*10^7 t SO_2-eq. Fig. 2(c) shows the acidification potential of crop farming was firstly higher than that of livestock breeding, while the former became lower than the latter after 1995. The reasons of this is the same as that causing N_2O, which has been illustrated above.

Besides NH_3 and NO_x, NO_3 and P loss are also determined by the fertilizer application, especially the applied manure excreted by the livestock. Thus, all these emissions contributing eutrophication had the same trends as those caused acidification. Similarly, as shown in Fig. 2(d), the eutrophication potential of agricultural production increased quickly to near three times from 2.59*10^7 t PO_4^{3-}-eq, though the eutrophication potential of crop farming began to decrease after 1990s with the fertilizer application being decreased. Because of the expanding of livestock breeding, a great deal of manure is discharged and emits many nutrients to the water environment. This has also caused the high increase of eutrophication potential of livestock breeding, which skipped from 3.52*10^6 t PO_4^{3-}-eq to 4.55*10^7 t PO_4^{3-}-eq during these 37 years.

3.2. Comparing sub-categories

As illustrated, the nutrient-derived environmental impacts are calculated for a total of nine different crop categories and four different livestock categories. The results are presented for these crop and livestock sub-categories. Then, each sub-category from all the countries is summed to that of the national total. For analyzing the environmental impacts in detail and providing more effective impact-mitigation strategies, the study compares the impacts between these sub-categories.

Fig. 5 shows the contribution of each crop and livestock for the energy consumption. It can be seen the three crops including rice, wheat, and maize consumed the much higher energy than other crops and livestock. As the three main crops including rice, wheat, and maize occupied the most yields in China, and the use-intensities of the synthetic N fertilizer are also much higher than other crops including those cash crops (eg. vegetables and fruits) (PDNDRC, 2001, 2003, 2006, 2011, 2016), they consumed the largest synthetic N fertilizer. With the high fertilizer use, these crop harvests also grew fast. However, with the raise of farmers' income, they are likely purchasing and using an increasing amount of agricultural materials especially the fertilizer, but are receiving no obvious improvement in the net economic benefit as a result (Zhang et al., 2012). It also can be seen the energy consumed by

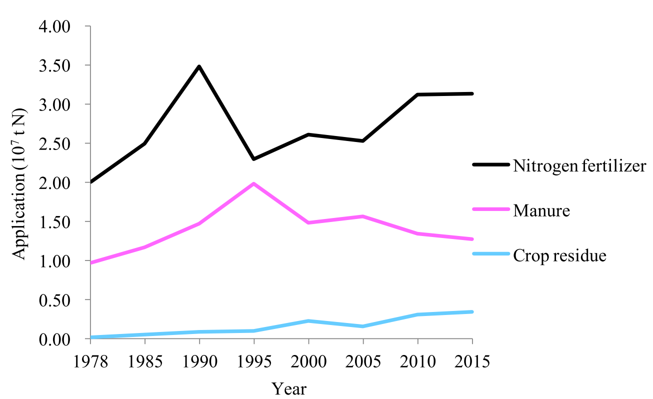

Fig. 4. N contents of the N applications in crop farming of China during 1978–2015.

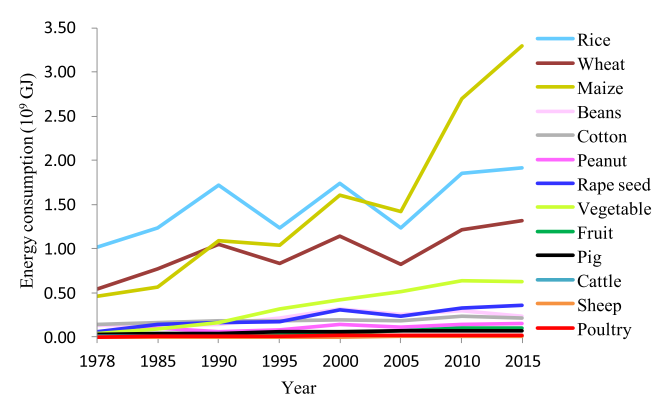

Fig. 5. Contribution of each crop/livestock to the energy consumption potential.

maize began to grow rapidly after 2005, which was far beyond all other sub-categories. This was mainly due to the great increase of harvest of maize, and its related residue applied to field. After 1995, the fertilizer application on vegetable grew abruptly. Thus, vegetable also consumed great energy from organic fertilizer in recent years only next to those three crops. Yan et al. (2016) estimated that over 50% of the total manure generated in the livestock industry was annually recycled in vegetable planting in China. While other crops such as oil crops (rapeseed, beans, cotton, peanut) and fruit in turn consumed the least energies, due to their much lower fertilizer-use intensities. Concerning the livestock, pig consumed the much higher energy than other three livestock. Because the energy equivalent of feed for pig is nearly 10 times than poultry (Table S2), and the amount of pig also increased fast. Contrarily, though the amount of sheep grew during these years, it was much smaller than other three livestock. Moreover, the energy equivalent of feed for sheep is only 1/8 of the pig. Thus, the sheep contributed the smallest energy consumption.

As noted in the model, the climate change is mainly caused by the nitrogen fertilizer, applied manure, and applied crop residue. In addition, the three main crops have the largest croplands and consumed the three greatest nitrogen fertilizers. The farmers also used the fertilizers inefficiently as discussed above. The high inputs and low use-efficiencies of resources in these staple crops promoted the deterioration of the climate (Chen et al., 2014). Fig. 6 illustrates these three crops especially rice overall contributed much more climate change than the oil crops, fruit, and livestock. While with the governments and scientists' growing attention on environmental protection after 1990, the fertilizer use-intensity of these main crops decreased, which alleviated the climate change. As the climate change relates closely with the energy consumption, the vegetable also contributed the nonnegligible climate change potential, which also increased after 1995. Besides these staple crops, pig emitted much higher N_2O, even exceeded all other crops after 2005. Poultry breeding also emitted much N_2O. As the meat consumption increased with the improving economy and the changing diet, livestock production has been flourishing. Among these livestock, the amounts of pig and poultry increased faster. Especially the traditional household feeding style of rising a few chicken and pigs has been replaced by the corporations housing thousands of livestock to supply the rising market demands (Zhou et al., 2007). With the amounts of these livestock increased rapidly in recent years, the excreted manure also increased faster, which emitted much N_2O (IPCC, 2007).

It has been illustrated the acidification is caused by the emissions of NH_3 and NO_x, both of which are determined by the excreted manure in livestock breeding. Thus, the acidification trends of these sub-categories and the reasons were similar as those of climate change (Fig. 7). While it should also notice the acidification resulted from pig had the same trend as the climate change. It also grew rapidly especially after 1990, exceeding the acidification from rice as the greatest contributor. While other crops including wheat and maize contributed much acidification, only next to pig and rice. This was mainly due to their large croplands and the great fertilizer applications as discussed.

It could be found in Fig. 8 that rice caused the much higher eutrophication than all other crops and livestock before 2000, due to its high fertilizer use, sown area, and harvests. While the sown area and fertilizer use-intensity decreased obviously after 1990, which also contributed the slow increase of the total eutrophication of agricultural production. Unlike the contributions to other three impact categories, other two staple crops including wheat and maize contributed the much lower eutrophication potential than the livestock. This maybe mainly due to the less P loss from these crops than the livestock. The eutrophication caused by vegetable is also obvious, even almost exceeded other crops and livestock after 2000. This was mainly related to its high application of synthetic fertilizer and no straw generated. In addition, all the eutrophication potentials caused by pig, cattle, poultry, and sheep nearly increased during these 37 years, because of the rising demand. Concerning the livestock breeding, the eutrophication potentials caused by pig and cattle were similar, and contributed more eutrophication potential than poultry and sheep. This mainly related with the population and N excretion of each livestock, as shown in the model. Because the population of pig was large, and its N excretion was also more than those from sheep and poultry. In addition, the N

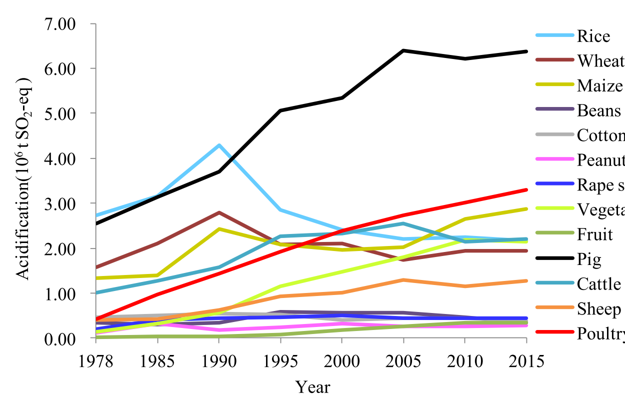

Fig. 7. Contribution of each crop/livestock to the acidification potential.

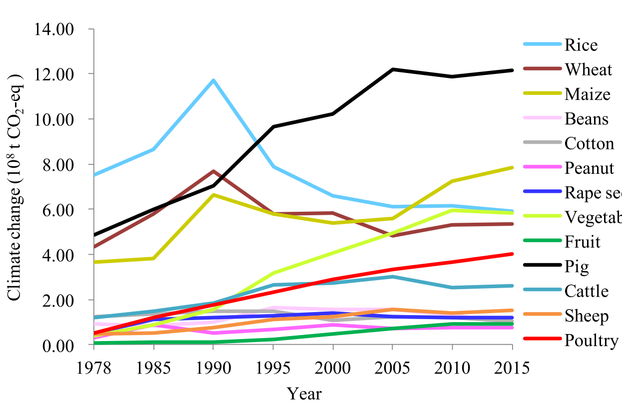

Fig. 6. Contribution of each crop/livestock to the climate change potential.

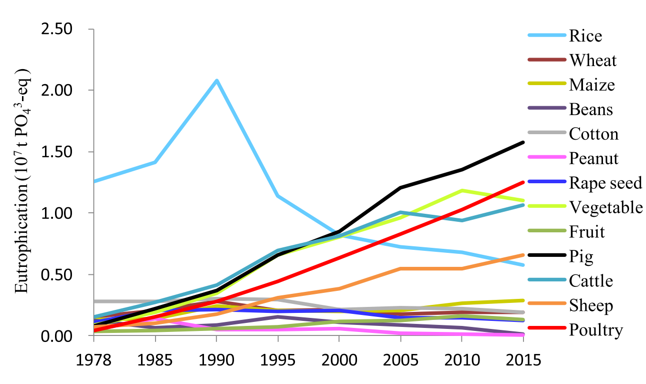

Fig. 8. Contribution of each crop/livestock to the eutrophication potential.

excretion per cattle was much higher than other three livestock. Meanwhile, the population of poultry was in fact much more than other livestock, though the N excretion per poultry was much less.

For verifying some similarities and dissimilarities between these categories, the study also utilizes the cluster analysis. Cluster analysis is a data mining process which consists in dividing the samples into groups (clusters) based on information found within the data which describes these samples and its relationships (Tan et al., 2006). These grouped samples show a similarity pattern while being as dissimilar as possible from samples associated to other clusters. Firstly, the study settles the four environmental impacts as the indexes, according to the 13 categories. For simplicity, the study also selects the average value during 1978–2015 of every index (Table 2).

Fig. 9 is the dendrogram which is the graphical representation of the clustering. The y-axis represents the 13 categories, and the x-axis denotes the relative distance of the categories. As illustrated in Table 2, each number represents a category. It could be seen the clustered results are mainly accordance with the above analysis. Firstly, there are two main classes identified. One contains 6, 9, 4, 7, 5, 12, 11, and 13, and the other contains 8, 2, 3, 1, and 10. As discussed above, oil crops (peanut, beans, and rape seed), fruit, cotton, cattle, and sheep resulted in the much lower fertilizer-use intensities, the energy equivalent of feed, and the N excretion, which contributed less environmental impacts than vegetable, wheat, maize, rice, and pig. Furthermore, in the former group, 6, 9, 4, 7, 5, 12, and 11 are more similar than 13. This is mainly due to the poultry's increasing number and manure, causing nonnegligible substances especially N_2O. Though the latter group overall contributed much more environmental impacts than the former, rice and pig generated the most obvious impacts. This is accordance with their high quantities, use of N fertilizer, and discharged manure, which has also been illustrated.

3.3. Comparing study sites

When compared with other studies researching the main N-related emissions from agriculture in China, as illustrated in Table 3, these emissions had no much difference with each other. It is noted Cui et al. (2013a) estimated more N_2O, N_2 and NH_3 than this study, mainly because they not only considered more crops and livestock, but also included residents living. While these emissions in this study were close to Gu et al. (2015), which defined the farming and breeding as the main agricultural activities, and analyzed the similar nutrient flows in these subsystems. It should also mention the N runoff and leaching from crop farming in this study was obviously higher than other studies, as we considered most of the applied fertilizer (chemical fertilizer, manure, and crop residue)

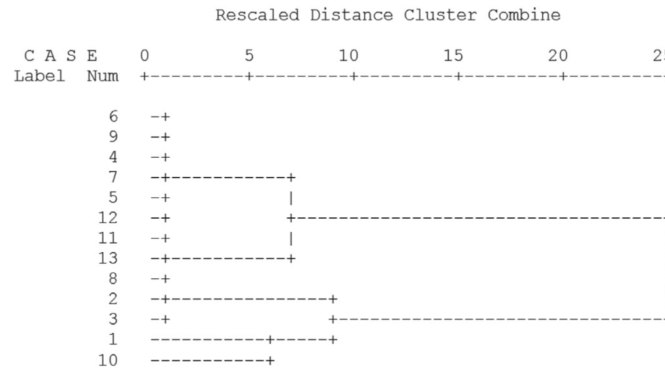

Fig. 9. Dendrogram for discriminating the categories in Chinese agriculture.

was emitted to the water environment.

As shown in Table 4, the total N-related emission from agriculture in China accounted for 66.0% in the world and 78.0% in the developing countries. Comparing these emissions to some developed countries/regions such as United States and EU27, all of these emissions in China were almost two times higher. In addition, the N-related emission per unit area was even almost 20 times higher than the word average. In detail, N_2O, NH_3, and N leaching occupied 9–39%, 30–99%, 42%, and 66% of the totals from agricultural in the world, and the emissions per unit area were 2–20 times higher than the world average. These were mainly due to several reasons. Firstly, as China produces and consumes the most synthetic N fertilizer in the world, the Chinese agriculture emitted a significant N-related losses. Meanwhile, with contrast to other regions such as developed countries, Chinese farmers apply fertilizer to millions of small plots, often resulting in gross overapplication (Ju et al., 2009). Moreover, the Chinese diet shifting to meat has stimulated a dramatic increase of livestock production, especially the intensive livestock production systems, and with that the manure production (Li et al., 2014). In addition, the lack of proper environmental infrastructure in the rural areas of China leads to the direct discharge of the wastewater and wastes to the environment, which also causes the deterioration of water quality in China (Chen and Chen, 2008).

Concerning N_2O emitted from the whole cycle of the agricultural production, and contributed entirely to the global warming, many studies researched N_2O from agricultural production and compared the contribution of each sub-category to N_2O. For example, Ma et al. (2014b) indicated the total N_2O from the crop farming more than doubled in China between 1980 and 2005, the trend of which is similar as this study. They also showed the N fertilizer application contributed over 50% to the direct emission of N_2O. This is also

Table 2
Type and description of the descriptive variables.

Number	Type	Energy consumption (10^9 GJ)	Climate change (10^8 t CO_2-eq)	Acidification (10^6 t SO_2-eq)	Eutrophication (10^7 PO_4^{3-}-eq)
1	Rice	1.50	7.57	2.76	1.09
2	Wheat	0.96	5.62	2.04	0.20
3	Maize	1.52	5.75	2.09	0.21
4	Beans	0.22	1.23	0.44	0.09
5	Cotton	0.19	1.27	0.46	0.25
6	Peanut	0.10	0.69	0.25	0.05
7	Rape seed	0.22	1.15	0.42	0.17
8	Vegetable	0.35	3.35	1.22	0.67
9	Fruit	0.05	0.45	0.16	0.09
10	Pig	0.06	9.25	4.85	0.79
11	Cattle	0.01	2.26	1.92	0.67
12	Sheep	0.00	1.08	0.89	0.35
13	Poultry	0.01	2.46	2.02	0.58

Table 3
Comparison of the main N-related emissions from Chinese agriculture in different studies (Tg N).

Source [year]	Subsystem	Item				
		N_2O	N_2	NH_3	NO_x	Runoff and leaching
Cui et al. (2013a) [1978]	Crop farming	4.70		5.10		5.90
	Livestock breeding	–	–	–	–	–
Gu et al. (2015) [1980]	Crop farming	0.20	3.40	3.10	0.10	1.90
	Livestock breeding	–	–	1.70	–	1.3
This study [1978]	Crop farming	0.41	2.70	3.00	0.04	16.80
	Livestock breeding	0.15	1.51	1.90	0.04	0.04
Xing and Zhu (2002) [1995]	Crop farming	5.00–10.00		6.10		11.00
Cui et al. (2013a) [1995]	Crop farming	9.00		9.60		10.00
This study [1995]	Crop farming	0.60	3.93	4.37	0.06	21.30
Cui et al. (2013a) [2010]	Crop farming	10.00		11.00		9.90
	Livestock breeding	–	–	–	–	–
Gu et al. (2015) [2010]	Crop farming	0.40	7.90	7.70	0.40	4.50
	Livestock breeding	–	–	4.60	–	3.30
Chen et al. (2016) [2011]	Crop farming	0.70		4.90		4.80
	Livestock breeding	–	–	–	–	–
This study [2010]	Crop farming	0.64	4.29	4.76	0.06	20.00
	Livestock breeding	0.42	4.16	5.42	0.11	0.11

Table 4
Comparison of the main N-related emissions from agriculture between China and other regions.

Unit	Region/Country	Item		
		N_2O	NH_3	Runoff and leaching
Total (Tg N)	China	1.06	10.20	20.10
	World[a]	2.70 −11.50	10.30 −34.20	47.80
	Developing countries[a]	1.90	23.80	14.70
	United States[b]	0.80	3.10	9.00
	EU27[c]	0.70	3.20	10.50
Per unit area (kg N ha^{-1})	China	4.72	35.00	147.00
	World[a]	0.50–2.30	1.70–6.90	9.70

[a] Beusen et al. (2008), Chen et al. (2016), Eickhout et al. (2006), FAOSTAT (2015), Galloway and Cowling (2002), Snyder et al. (2009).
[b] Doering et al. (2011).
[c] Leip et al. (2011).

similar as the contribution of N fertilizer application to the direct emission of N_2O from crop farming in this study, which was average of 63.0%. Regarding the emission of N_2O in the livestock breeding in China, Zhou et al. (2007) found the N_2O from manure management in China was $4.78*10^4$ t in 1949, increased to $2.41*10^5$ t in 2003, with an annual growth rate of 3.0%, most of which was contributed by pig. Similarly, pig contributed the most N_2O in this study. While the N_2O emitted from livestock breeding in China during 1978–2015 increased from $2.37*10^5$ t to $6.81*10^5$ t, with an annual growth rate of 5.0%. This is mainly due to the dramatic increase of livestock since the mid-1990s in China, and the inadequate manure management practices especially in the breeding enterprises, which exist in large numbers in China. The contributions of each sub-category to N_2O could be also compared to other countries. Wang et al. (2011) showed grassland and crops were the major contributors (42.0% and 22.0%) to N_2O emission in UK, though which were smaller than the contribution of crops to China in this study (57.0%–74.0% during 1978–2015). This were mainly due to the large cropland and high fertilizer application in China. Concerning the livestock, cattle contributed much more N_2O than other livestock including pig, sheep, and poultry in UK (Wang et al., 2011). This conclusion was totally different with this study, which found pig contributed 3–10 times N_2O than other three livestock in China. This maybe mainly because of the rapid growth of pig, resulted from the Chinese high population and diet transition to meat especially pork.

3.4. Implication for sustainable nutrient management

3.4.1. Improve crop production

3.4.1.1. Limit nitrogen fertilizer especially on rice, wheat, and maize. As the high applications of the nitrogen fertilizer on the crops especially the three main crops, which affected obviously on all the environmental impacts, it is important to limit the nitrogen fertilizer on these crops. Chen et al. (2016) estimated the N productivity in China would decrease slightly by 1.1% for the whole agricultural production system and 2.1% the crop farming system, when the application of nitrogen fertilizer is reduced by 30% from its 2011 level. This means the N-related emissions could be reduced by 15.6%, with NH_3, N_2O, leaching nitrate, and runoff TN declined by 11.5%, 16.8%, 21.0%, and 14.2%, respectively. Koubouris et al. (2017) also suggested decreasing the carbon inputs and the irrigation in soil layer 0–10 cm would limit NO_3–N and NH_4–N. With regard to the three staple crops, Xia et al. (2016) shown the total amount of N fertilizers applied for rice should be split into at least three applications: basal fertilization, early tiller, panicle initiation and heading stages. There are also some studies noted implementing two top-dressings (with one topdressing applied at the later growth stage) on wheat and corn had the priority on the greatly increase yields and the reduction of N application rates, compared to the current one top-dressing practice (Zhang et al., 2012; Cui et al., 2013b; Chen et al., 2014).

3.4.1.2. Enlarge the use of crop residue and manure. The study showed the energy embodied in crop residue grows rapidly even highly exceeded other N agri-resources. Though some researchers found that the return of crop residue such as straws and pruning residues to the field is effective to reduce the impact resulted from the chemical fertilizer (Krenitsky et al., 1998; Jin et al., 2009) or increases the content of organic matter in the soil (Repullo et al., 2012). Kim Oanh et al., 2005 found the incomplete combustion of many straws would lead to the emissions of a large amount of pollutants, which would also bring serious health threats to the local residents. In fact, the open burning of straws is widely implemented in China and has caused serious air pollution. Moreover, the N content of applied manure is only lower than nitrogen fertilizer, which also affects directly on the N-related emissions and corresponding environmental impacts. Thus, it is

urgent to reduce the application of crop residue and manure to field and enlarge the use for other purposes. For example, the manure could be used in biogas production or as a feedstock for fish besides being applied as a fertilizer. Ma et al. (2014b) also given some recommendations on manure management, such as giving subsidies for building manure storages, and penalizing manure discharges. Furthermore, it is fortunate that the Ministry of Agriculture of China has promulgated the Action Plan to *Achieve Zero-growth of Synthetic Fertilizer Application by 2020*, which aims to increase the recycling rates of animal manure and crop residue both to 60% by 2020.

3.4.1.3. Adjust the crop structure and focus more on oil crops and fruits. The results showed the fruit, oil crops (beans, cotton, peanut, and rapeseed), and vegetable contributed lower environmental impacts than the grain crops, except for vegetable contributing more eutrophication resulted from its high application of synthetic fertilizer and no straw generated. Thus, the government should not only provide an incentive for farmers to cultivate more fruit, oil crops, and vegetable, but also guide them to gradually reduce the application rates of chemical fertilizer, and increase their adoption of knowledge-based application methods. In fact, in April 2016, the Ministry of Agriculture announced the *Adjustment Plan of National Planting Structure (2016–2020)*. This plan has not only highlighted guarantying the grain crops and cotton, restoring the area of soybean planting, focusing on planting of peanut and rapeseed, improving the quality of vegetable production, but also advanced the different planting focus of different regions in China.

3.4.2. Improve livestock production

3.4.2.1. Control breeding of pig and poultry. As discussed above, pig and poultry especially pig contributed the most to the environmental impacts than other livestock, due to the huge amounts and manure. Hence, it is suggest to limit the breeding of these two livestock, which could be achieved by the following measures. Firstly, the government should prohibit or limit breeding these livestock especially in the water source areas, living areas, and scenic spots. Then the government should also provide the breeding enterprises with some funds and technical guidance to recycle and harmlessly dispose the manure. Meanwhile, the breeding enterprises must improve their waste disposal infrastructure, and spread the knowledge of environmentally conscious practices.

3.4.2.2. Encourage the breeding of sheep. Sheep breeding generated more less N_2O and NH_3 than other livestock, thus it is reasonable to expanding the breeding scale of sheep. In fact in 2016, China has given subsides to encourage the sheep breeding enterprises or individuals breeding over 50 sheep. In addition, breeding enterprises or individuals need to study some technologies of breeding sheep, including seed selection, feed provision, disease prevention, manure management, etc.

3.4.2.3. Improve the manure treatment. The capabilities of disposing manure should be improved. Firstly, the local governments should strengthen the supervision of the waste disposal infrastructure in breeding enterprises, with some penalties and rewards. Moreover, the breeding enterprises are suggested to improve the infrastructure of waste disposal, study some knowledge or techniques of eco-breeding, and clean the breeding areas frequently. Furthermore, the breeder could substitute some crops and crop residues for the chemical feed, which could not only reduces nutrient uptake but also decreases the feed cost. Like Mahan and Howes (1995) early showed the dietary P inputs for livestock could be matched to the requirements of livestock to decrease the amount of P loss.

3.4.3. Change dietary pattern

Considering the different environmental impacts from different crops and livestock, to establish a healthy diet and improving lifestyle are essential. There are also many studies (Kummu et al., 2012; Westhoek et al., 2014) stated reducing the intake of animal-derived protein would reduce demand and consequently the production of these food types, thereby decreasing the associated N_2O emissions. The residents should not only reduce the intake of animal-derived foods especially pig- and poultry-derived foods, but also supplement with consuming more vegetables, fruits and oil crops. This changed dietary pattern would not reduce the residents' nutrient absorption, as Wu et al. (2015) found the P consumed by both rural and urban residents in China during 1980–2012 were higher than at least 2.5 times than that of the Recommended Dietary Allowance (RDA). Early Smil (2007) also suggested that shifting to a 'smart vegetarian' diet in combination with reducing over-consumption is one of the most cost-effective measures in reducing agricultural resource inputs. Moreover, the residents should minimize the food waste, which was estimated up to 60 million tons and the increasing rate of its production was higher than 10% per year due to the growth of population and rising of living standards (Li et al., 2013; Zhang et al., 2016c).

3.5. Uncertainty and limitation

There are a large number of sources from which uncertainty arises. Firstly, the statistical data especially the data before 2000 have intrinsic uncertainties, which are difficult to assess without further information. Secondly, the nutrient contents and emission factors were derived from various literature sources, which are needed to verify these estimates. Moreover, some data such as the energy equivalents, proportions of manure and straw applied to field maybe different between the different crops and livestock. While these data actually could hardly obtained clearly basing on the different crops and livestock. Meanwhile, although there was a detailed category of cops and livestock in the inventory of estimating environmental impacts, the inventory did fail to capture some natural factors, such as temperature, pH, and wind speed. These factors related to metrological conditions and seasonal trends in agricultural practices will affect some emissions as well (Wang et al., 2011).

Moreover, the studied environmental impacts are mainly assessed by multiplying their corresponding environmental indicators with the characterization factors. Some quantifications and sources of these data may have some uncertainties as discussed above. Meanwhile, many data are not differentiated basing on the sub-regions in China, while different sub-regions may show different characteristics. Kourgialas et al. (2017) proposed a geological information system (GIS) policy approach for assessing the effect of fertilizers on the quality of drinking and irrigation water and wellhead protection zones. They concerned some pollutants including major- and trace-elements related to the fertilizers. The pollutant concentrations were obtained from monitoring the groundwater sample in different wells. From the available data and by using GIS technique, the spatial distribution of the pollutant concentrations in the groundwater were estimated. This approach could be referred to assess more credibly some environmental impacts such as the acidification and the eutrophication, if some spatial data could be obtained by some techniques such as monitoring and GIS.

In addition, the study not only analyzes the different impacts from different subsystems (farming and breeding), but also compares the results between different subcategories (crops and

livestock), which made the calculation relatively large. Thus the study only selects crop farming and livestock breeding, and does not consider other processes, such as agricultural resource production, crop processing, residents consumption, and waste treatment. While these processes are also the parts of the life cycle of agricultural production (Wu et al., 2014, 2015), and the nutrients also flow through them and result in some environmental impacts (Fernandez-Mena et al., 2016; Wu et al., 2016; Zhang et al., 2013). Correspondingly, it would be more efficient to combine more processes so that more holistic views can be achieved.

Besides these, the study analyzes those environmental impacts during 1978–2015, while only selects years of 1978, 1985, 1990, 1995, 2000, 2005, 2010, and 2015 for detailed calculation due to data availability. The results may change annually, though there are many related studies being analyzed with a five-even ten-year period (Cui et al., 2013a; Ma et al., 2014b; Zhang et al., 2013; Zhou et al., 2007). The future studies should consider additional data collection and accurate calculations so that a more complete and more accurate results can be expected. Meanwhile, this study presents an overall picture of the system in China, and ignores the regional variations limited by the workload. Actually, the nutrient-derived environmental impacts may vary geospatially resulted from the variable agricultural structure, production style, natural environment, economic development, etc. The results would be more meaningful if the study subdivided the country into smaller units such as provinces.

4. Conclusions

By combining LCA method with the mass balance principle of SFA, the study constructs the NEIA model to analyze the dynamic characteristics of the environmental impacts caused by nutrients in the agricultural production of China. It shows all the environmental impacts including energy consumption, global warming, acidification, and eutrophication had increased during 1978–2015, which had increased to $8.22*10^9$ GJ, $5.01*10^8$ t CO_2-eq, $2.41*10^7$ t SO_2-eq, and $7.18*10^7$ t PO_4^{3-}-eq, respectively. Moreover, crop farming contributed average 60 and two times energy consumption and climate change as livestock breeding. While the latter subsystem exceeded the former subsystem in acidification and eutrophication after 1995 and 2000, which were similar with each other. This was mainly due to the high N application including synthetic N fertilizer (from $1.33*10^9$ GJ to $2.08*10^9$ GJ), applied manure (from $4.94*10^8$ GJ to $5.65*10^8$ GJ), and applied crop residue (from $2.94*10^8$ GJ to $5.30*10^9$ GJ), which embodied much more energies and also emitted many nutrients to environment. On the other hand, after the mid of 1990s, the synthetic N fertilizer was controlled under the government's highlight on environmental protection, and the livestock expanded rapidly with people's diet transition to meat consumption, this lead to the breeding generated nonnegligible environmental effects.

Concerning the environmental impacts among the different crops and different livestock, the three crops including rice, wheat, and maize almost contributed about two to 10 times environmental impacts as other crops and livestock, due to their high fertilizer uses, sown areas, and harvests. Contrarily the oil crops and fruit consumed the least energies because of their much lower fertilizer uses. On the other hand, influenced by the feeding quantity and excretion, the pig and the poultry especially the pig emitted much higher N_2O and P loss, which contributed almost two to 20 times climate change, acidification, and eutrophication as other livestock.

Based on the characteristics of these environmental impacts, the study presents the nutrient management for sustainable agriculture, such as limiting nitrogen fertilizer on Chinese three staple crops, enlarging the use of crop residue and manure, cultivating more oil crops and fruits, controlling the breeding of pig and poultry, improving manure treatment, and reducing the residents' intake of animal-derived food.

In general, this study can provide valuable experiences on mitigating environmental impacts from agricultural sector to other regions. In addition, this study provides model and relevant data inventories for similar studies in the future. Future work should improve and assess the data quality, involve more nutrient-related food system, and analyze the impacts on larger temporal and spatial scales.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jenvman.2018.04.002.

References

Asmala, E., Saikku, L., Vienonen, S., 2011. Import-export balance of nitrogen and phosphorus in food, fodder and fertilizers in the Baltic Sea drainage area. Sci. Total Environ. 409, 4917–4922.
Baker, L.A., Hope, D., Xu, Y., Edmonds, J., Lauver, L., 2001. Nitrogen balance for the Central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4 (6), 582-60.
Beusen, A.H.W., Bouwman, A.F., Heuberger, P.S.C., Van Drecht, G., Van Der Hoek, K.W., 2008. Bottom-up uncertainty estimates of global ammonia emissions from global agricultural production systems. Atmos. Environ. 42 (24), 6067–6077.
Bouwman, L., Goldewijk, K.K., Van Der Hoek, K.W., Beusen, A.H.W., Van Vuuren, Willems, J., Rufino, M.C., Stehfest, E., 2013. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. Proc. Natl. Acad. Sci. 110 (52), 20882–20887.
Brentrup, F., Küsters, J., Kuhlmann, H., Lammel, J., 2004a. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology I. Theoretical concept of a LCA method tailored to crop production. Eur. J. Agron. 20, 247–264.
Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., Kuhlmann, H., 2004b. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur. J. Agron. 20 (3), 265–279.
Brunner, P.H., 2010. Substance flow analysis as a decision support tool for phosphorus management. J. Industrial Ecol. 14 (6), 870–873.
Canfield, D.E., Glazer, A.N., Falkowski, P.G., 2010. The evolution and future of Earth's nitrogen cycle. Science 330 (6001), 192–196.
Chen, M., Chen, J., Sun, F., 2008. Agricultural phosphorus flow and its environmental impacts in China. Sci. Total Environ. 405 (1–3), 140–152.
Chen, M., Chen, J., Sun, F., 2010. Estimating nutrient releases from agriculture in China: an extended substance flow analysis framework and a modeling tool. Sci. Total Environ. 408 (21), 5123–5136.
Chen, M., Chen, J., 2008. Phosphorus release from agriculture to surface waters: past, present and future in China. Water Sci. Technol. 57 (9), 1355–1361.
Chen, M.P., Graedel, T.E., 2016. A half-century of global phosphorus flows, stocks, production, consumption, recycling, and environmental impacts. Glob. Environ. Change 36, 139–152.
Chen, M.P., Sun, F., Shindo, J., 2016. China's agricultural nitrogen flows in 2011: environmental assessment and management scenarios. Resour. Conservation Recycl. 111, 10–27.
Chen, X.P., Cui, Z.L., Fan, M.S., et al., 2014. Producing more grain with lower environmental costs. Producing more grain with lower environmental costs. Nature 514 (7523), 486–489.
Chen, Z.L., Tang, Y.Z., 1999. Study on sustainable use of urban night-soil in China. Urban Environ. Urban Ecol. 12 (2), 42–49.
Cui, S.H., Shi, Y.L., Groffman, P.M., Schlesinger, W.H., Zhu, Y.G., 2013a. Centennial-scale analysis of the creation and fate of reactive nitrogen in China (1910–2010). Proc. Natl. Acad. Sci. 110 (6), 2052–2057.
Cui, S.H., Xu, S., Huang, W., Bai, X.M., Huang, Y.F., Li, G.L., 2015. Changing urban phosphorus metabolism: evidence from Lonyan city, China. Sci. Total Environ. 536, 924–932.
Cui, Z., Yue, S., Wang, G., Zhang, F., Chen, X., 2013b. In-season root-zone N management for mitigating greenhouse gas emission and reactive N losses in

intensive wheat production. Environ. Sci. Technol. 47, 6015–6022.
de Vries, W., Kros, J., Dolman, M.A., Vellinga, T.V., de Boer, H.C., Gerritsen, A.L., Sonneveld, M.P.W., Bouma, J., 2015. Environmental impacts of innovative dairy farming systems aiming at improved internal nutrient cycling: a multi-scale assessment. Sci. Total Environ. 536, 432–442.
Doering, O., Galloway, J.N., Theis, T.L., Aneja, V., Boyer, E., Cassman, K.G., Cowling, E.B., Dickerson, R.R., Herz, W., Hey, D.L., Kohn, R., Lighy, J.S., Mitsch, W., Moomaw, W., Mosier, A., Paerl, H., Shaw, B., Stacey, P., 2011. Reactive Nitrogen in the United States: an Analysis of Inputs, Flows, Consequences, and Management Options. United States Environmental Protection Agency, Washington, DC.
Devendra, C., 2007. Perspectives on animal production systems in Asia. Livest. Sci. 106 (1), 1–18.
Eickhout, B., Bouwman, A.F., Van Zeijts, H., 2006. The role of nitrogen in world food production and environmental sustainability. Agric. Ecosyst. Environ. 116 (1), 4–14.
Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z., Winiwarter, W., 2008. How a century of ammonia synthesis changed the world. Nat. Geosci. 1, 636–639.
FAOSTAT, 2015. United Nations Food and Agriculture Organizations (FAO). Statistical Database.
Fei, R.L., Lin, B.Q., 2016. Energy efficiency and production technology heterogeneity in China's agricultural sector: a meta-frontier approach. Technol. Forecast. Soc. Change 109, 25–34.
Fernandez-Mena, H., Nesme, T., Pellerin, S., 2016. Towards an Agro-Industrial Ecology: a review of nutrient flow modelling and assessment tools in agro-food systems at the local scale. Sci. Total Environ. 543, 467–479.
Foley, J.A., 2011. Can we feed the world and sustain the planet. Sci. Am. 305 (5), 60–65.
Foteinis, S., Chatzisymeon, E., 2016. Life cycle assessment of organic versus conventional agriculture. A case study of lettuce cultivation in Greece. J. Clean. Prod. 112, 2462–2471.
Fowler, D., Coyle, M., Skiba, U., et al., 2013. The global nitrogen cycle in the twenty-first century. Phil. Trans. R. Soc. B Biol. Sci. 368 (1621), 20130164.
Fu, B.J., Zhuang, X.L., Jiang, G.B., Shi, J.B., Lu, Y.H., 2007. Feature: environmental problems and challenges in China. Environ. Sci. Technol. 41, 7597–7602.
Galloway, J.N., Cowling, E.B., 2002. Reactive nitrogen and the world: 200 years of change. AMBIO A J. Hum. Environ. 31 (2), 64–71.
Garrido, R., Silvestre, J.D., Flores-Colen, I., 2017. Economic and energy life cycle assessment of aerogel-based thermal renders. J. Clean. Prod. 151, 537–545.
Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food security: the challenge of feeding 9 billion people. Science 327 (5967), 812–818.
Gu, B.J., Ju, X.T., Chang, J., Ge, Y., Vitousek, P.M., 2015. Integrated reactive nitrogen budgets and future trends in China. Proc. Natl. Acad. Sci. 112 (28), 8792–8797.
Guinée, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. Int. J. Life Cycle Assess. 7, 311–313.
Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P., Goulding, K.W.T., Vitousek, P.M., Zhang, F.S., 2010. Significant acidification in major Chinese croplands. Science 327, 1008–1010.
Hamilton, H.A., Brod, E., Hanserud, O., Müller, D.B., Brattebø, H., Haraldsen, T.K., 2017. Recycling potential of secondary phosphorus resources as assessed by integrating substance flow analysis and plant-availability. Sci. Total Environ. 575, 1546–1555.
Hasler, K., Bröring, S., Omta, S.W.F., Olfs, H.W., 2015. Life cycle assessment (LCA) of different fertilizer product types. Eur. J. Agron. 69, 41–51.
Huijbregts, M.A.J., 2001. Uncertainty and Variability in Environmental Life-cycle Assessment. PhD thesis. University of Amsterdam, Amsterdam.
IPCC, 2007. Climate change 2007: mitigation of climate change. In: Metz, B., Davidson, O.R., Bosh, P.R., Dave, R., Meyer, L.A. (Eds.), Contribution of the Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom.
ISO E, 2006. Environmental management-Life cycle assessment-Principles and framework, 14040 Eur. Comm. Stand. 2006.
Jiang, S.Y., Yuan, Z.W., 2015. Phosphorus flow patterns in the Chaohu Watershed from 1978 to 2012. Environ. Sci. Technol. 49 (24), 13973–13982.
Jin, K., Cornelis, W.M., Schiettecatte, W., Lu, J.J., Cai, D.X., Jin, J.Y., De Neve, S., Hartmann, R., Gabriels, D., 2009. Effects of different soil management practices on total P and Olsen-P sediment loss: a field rainfall simulation study. Catena 78, 72–80.
Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., Cui, Z.L., Yin, B., Christie, P., Zhu, Z.L., Zhang, F.S., 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. Proc. Natl. Acad. Sci. U. S. A. 106 (9), 3041–3046.
Kahrl, F., Li, Y.J., Su, Y.F., Tennigkeit, T., Wilkes, A., Xu, J.C., 2010. Greenhouse gas emissions from nitrogen fertilizer use in China. Environ. Sci. Policy 13, 688–694.
Kim Oanh, N.T., Albina, D.O., Pign, L., Wang, X.K., 2005. Emission of particulate matter and polycyclic aromatic hydrocarbons from select cookstove-fuel systems in Asia. Biomass Bioenergy 28, 579–590.
Klepper, O., Beusen, A.H.W., Meinardi, C.R., 1995. Modelling the flow of nitrogen and phosphorus in Europe: from loads to coastal seas. Water Sci. Technol. 31, 141–145.
Klimont, Z., Brink, C., 2004. Modelling of emissions of air pollutants and greenhouse gases from agricultural sources in Europe. Environ. Effects Spacecr. Position. Traject. 131–137.
Koubouris, G.C., Kourgialas, N.N., Kavvadias, V., Digalaki, N., Psarras, G., 2017. Sustainable agricultural practices for improving soil carbon and nitrogen content in relation to water availability - an adapted approach to Mediterranean olive groves. Commun. Soil Sci. Plant Analysis 48 (22), 2687–2700.
Kourgialas, N.N., Karatzas, G.P., Koubouris, G.C., 2017. A GIS policy approach for assessing the effect of fertilizers on the quality of drinking and irrigation water and wellhead protection zones (Crete, Greece). J. Environ. Manag. 189, 150–159.
Krenitsky, E.C., Carroll, M.J., Hill, R.L., Krouse, J.M., 1998. Runoff and sediment losses from natural and man-made erosion control materials. Crop Sci. 38 (4), 1042–1046.
Kummu, M., De Moel, H., Porkka, M., Siebert, S., Varis, O., Ward, P.J., 2012. Lost food, wasted resources: global food supply chain losses and their impacts on freshwater, cropland, and fertilizer use. Sci. Total Environ. 438, 477–489.
Leip, A., Achermann, B., Billen, G., Winiwarter, W., 2011. Integrating nitrogen fluxes at the European scale. Eur. Nitrogen Assess. Sources Eff. Policy Perspect. 345–376.
Li, G., Li, H., Leffelaar, P.A., Shen, J., Zhang, F., 2014. Characterization of phosphorus in animal manures collected from three (dairy, swine, and broiler) farms in China. PLoS One 9 (7), e102698.
Li, Y., Zhang, R., Liu, X., Chen, C., Xiao, X., Feng, L., He, Y.F., Liu, G.Q., 2013. Evaluating methane production from anaerobic mono-and co-digestion of kitchen waste, corn stover, and chicken manure. Energy fuels. 27 (4), 2085–2091.
Lin, T., Wang, J., Bai, X.M., Zhang, G.Q., Li, X.H., Ge, R.B., Ye, H., 2016. Quantifying and managing food-sourced nutrient metabolism in Chinese cities. Environ. Int. 94, 388–395.
Linderholm, K., Tillman, A.M., Mattsson, J.E., 2012. Life cycle assessment of phosphorus alternatives for Swedish agriculture. Resour. Conservation Recycl. 66, 27–39.
Liu, X., Sheng, H., Jiang, S.Y., Yuan, Z.W., Zhang, C.S., Elser, J.J., 2016. Intensification of phosphorous cycling in China since the 1600s. Proc. Natl. Acad. Sci. 113 (10), 2609–2614.
Liu, Y., Villalba, G., Ayres, R.U., Schroder, H., 2008. Global phosphorus flows and environmental impacts from a consumption perspective. J. Industrial Ecol. 12, 229–247.
Liu, Y., 2004. Study on Phosphorus Societal Metabolism and Eutrophication Control Policy in China (PhD thesis). Tsinghua University, Beijing (in Chinese).
Luukkanen, J.J., Panula-Ontto, J., Vehmas, J., Liu, L.Y., Kaivo-oja, J., Häyhä, L., Auffermann, B., 2015. Structural change in Chinese economy: impacts on energy use and CO_2 emissions in the period 2013-2030. Technol. Forecast. Soc. Change 94, 303–317.
Ma, L., Guo, J.H., Velthof, G.L., Li, Y.M., Chen, Q., Ma, W.Q., Oenema, O., Zhang, F.S., 2014a. Impacts of urban expansion on nitrogen and phosphorus flows in the food system of Beijing from 1978 to 2008. Glob. Environ. Change 28 (1), 192–204.
Ma, L., Ma, W.Q., Velthof, G.L., Wang, F.H., Qin, W., Zhang, F.S., Oenema, O., 2010. Modeling nutrient flows in the food chain of China. J. Environ. Qual. 39 (4), 1279–1289.
Ma, L., Velthof, G.L., Kroeze, C., Ju, X.T., Hu, C.S., Oenema, O., Zhang, F.S., 2014b. Mitigation of nitrous oxide emissions from food production in China. Curr. Opin. Environ. Sustain. 9–10 (1), 82–89.
Ma, L., Velthof, G.L., Wang, F.H., Qin, W., Zhang, W.F., Liu, Z., Zhang, Y., Wei, J., Lesschen, J.P., Ma, W.Q., Oenema, O., Zhang, F.S., 2012. Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005. Sci. Total Environ. 434, 51–61.
Ma, L., Wang, F.H., Zhang, W.F., Ma, W.Q., Velthof, G.L., Qin, W., Oenema, O., Zhang, F.S., 2013. Environmental assessment of management options for nutrient flows in the food chain in China. Environ. Sci. Technol. 47 (13), 7260–7268.
Mahan, D.C., Howes, D., 1995. Environmental Aspects with Particular Emphasis on Phosphorus, Selenium, and Chromium in Livestock Feed. 13th Annual Pacific Northwest Animal Nutrition Conference, Portland, Oregon, USA.
Makowski, D., Nesme, T., Papy, F., Doré, T., 2014. Global agronomy, a newfield of research. A review. Agron. Sustain. Dev. 34, 293–307.
MOEP and MOA, 2010. China pollution Source Census (in Chinese). http://cpsc.mep.gov.cn/gwgg/.
Nabavi-Pelesaraei, A., Bayat, R., Hosseinzadeh-Bandbafha, H., Afrasyabi, H., Chau, K., 2017. Modeling of energy consumption and environmental life cycle assessment for incineration and landfill systems of municipal solid waste management-A case study in Tehran Metropolis of Iran. J. Clean. Prod. 148, 427–440.
National Bureau of Statistics of China (NBSC), 2016. China Statistical Yearbook. China Statistics Press, Beijing (in Chinese).
National Development and Reform Commission of the People (NDRC), 2012. Second National Communication on Climate Change of the People's Republic of China. United Nations Framework Convention on Climate Change (UNFCCC).
Nilsson, J., 1995. A phosphorus budge for Swedish municipality. J. Environ. Manag. 45 (3), 243–253.
Pearce, B.B.J., Chertow, M., 2017. Scenarios for achieving absolute reductions in phosphorus consumption in Singapore. J. Clean. Prod. 140, 1587–1601.
Price Department of the National Development and Reform Commission (PDNDRC), 2001. National Agricultural Products Cost-benefit Compilation. China Statistics Press, Beijing (in Chinese).
Price Department of the National Development and Reform Commission (PDNDRC), 2006. National Agricultural Products Cost-benefit Compilation. China Statistics Press, Beijing (in Chinese).
Price Department of the National Development and Reform Commission (PDNDRC), 2011. National Agricultural Products Cost-benefit Compilation. China Statistics

Press, Beijing (in Chinese).
Price Department of the National Development and Reform Commission (PDNDRC), 2016. National Agricultural Products Cost-benefit Compilation. China Statistics Press, Beijing (in Chinese).
Price Department of the National Development and Reform Commission (PDNDRC), 2003. National Agricultural Products Cost-benefit Compilation since Funding (1953-1997). China Prices Press, Beijing (in Chinese).
Repullo, M.A., Carbonell, R., Hidalgo, J., Rodríguez-Lizana, A., Ordóñez, R., 2012. Using olive pruning residues to cover soil and improve fertility. Soil Tillage Res. 124, 36–46.
Ridoutt, B.G., Wang, E., Sanguansri, P., Luo, Z., 2013. Life cycle assessment of phosphorus use efficient wheat grown in Australia. Agric. Syst. 120, 2–9.
Skiba, U., Fowler, D., Smith, K.A., 1997. Nitric oxide emissions from agricultural soils in temperate and tropical climates: sources, controls and mitigation options. Nutr. Cycl. Agroecosyst. 48, 139–153.
Smil, V., 2007. Policy for Improved Efficiency in the Food Chain, SIWI Seminar: Water for Food, Bio-fuels or Ecosystems? World Water Week. August 12th-18th 2007, Stockholm.
Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric. Ecosyst. Environ. 133 (3), 247–266.
Sutton, M.A., Bleeker, A., Howard, C.M., et al., 2013. Our Nutrient World: the challenge to Produce More Food and Energy with Less Pollution. NERC/Centre for Ecology and Hydrology.
Tan, P.N., Steinbach, M., Kumar, V., 2006. Introduction to Data Mining. Pearson Addison Wesley.
ten Hoeve, M., Hutchings, N.J., Peters, G.M., Svanström, M., Jensen, L.S., Bruun, S., 2014. Life cycle assessment of pig slurry treatment technologies for nutrient redistribution in Denmark. J. Environ. Manag. 132, 60–70.
Thitanuwat, B., Polprasert, C., Englande Jr., A.J., 2016. Quantification of phosphorus flows throughout the consumption system of Bangkok Metropolis, Thailand. Sci. Total Environ. 542 (Pt B).
Uwizeye, A., Gerber, P.J., Schulte, R.P.O., de Boer, I.J.M., 2016. A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains. J. Clean. Prod. 129, 647–658.
Van der Voet, E., 2002. SFA methodology. In: Ayers, R.U., Ayers, L.W. (Eds.), The Handbook of Industrial Ecology. Edward Elgar, Cheltenham, UK.
Vermeulen, S.J., Campbell, B.M., Ingram, J.S.I., 2012. Climate change and food systems. Annu. Rev. Environ. Resour. 37 (1), 195–222.
Vitale, P., Arena, N., Di Gregorio, F., Arena, U., 2017. Life cycle assessment of the end-of-life phase of a residential building. Waste Manag. 60, 311–321.
Wang, C.H., Liu, X.J., Ju, X.T., Zhang, F.S., 2002. In situ determination of ammonia volatilization from wheat maize rotation system field in north China. Acta Ecol. Sin. 22 (3), 359–365 (in Chinese).
Wang, J.Q., Ma, W.Q., Jiang, R.F., Zhang, F.S., 2007. Development and application of nitrogen balance model of agro-ecosystem in China (in Chinese). Transations Chin. Soc. Agric. Eng. 23, 210–215.
Wang, J.Y., Cardenas, L.M., Misselbrook, T.H., Gilhespy, H., 2011. Development and application of a detailed inventory framework for estimating nitrous oxide and methane emissions from agriculture. Atmos. Environ. 45 (7), 1454–1463.
Westhoek, H., Lesschen, J.P., Rood, T., Wagner, S., De Marco, S., Murphy-Bokern, D., Leip, A., van Grinsven, H., Sutton, M.A., Oenema, O., 2014. Food choices, health and environment: effects of cutting Europe's meat and dairy intake. Glob. Environ. Change 26, 196–205.
Wu, H.J., Gao, L.M., Yuan, Z.W., Wang, S., 2016. Life cycle assessment of phosphorus use efficiency in crop production system of three crops in Chaohu watershed. J. Clean. Prod. 139, 1298–1307.
Wu, H.J., Yuan, Z.W., Gao, L.M., Zhang, L., Zhang, Y.L., 2015. Life-cycle phosphorus management of the crop production–consumption system in China, 1980-2012. Sci. Total Environ. 502, 706–721.
Wu, H.J., Yuan, Z.W., Geng, Y., Ren, J.Z., Jiang, S.Y., Sheng, H., Gao, L.M., 2017. Temporal trends and spatial patterns of energy use efficiency and greenhouse gas emissions in crop production of Anhui Province, China. Energy 133, 955–968.
Wu, H.J., Yuan, Z.W., Zhang, L., Bi, J., 2012a. Eutrophication mitigation strategies: perspectives from the quantification of phosphorus flows in socioeconomic system at county level. J. Clean. Prod. 23 (1), 122–137.
Wu, H.J., Yuan, Z.W., Zhang, L., Bi, J., 2012b. Life cycle energy consumption and CO_2 emission of an office building in China. Int. J. Life Cycle Assess. 17 (2), 105–118.
Wu, H.J., Yuan, Z.W., Zhang, Y.L., Gao, L.M., Liu, S.M., 2014. Life-cycle phosphorus use-efficiency of the farming system in Anhui Province, Central China. Resour. Conservation Recycl. 83, 1–14.
Xia, L., Ti, C.P., Li, B.L., Xia, Y.Q., Yan, X.Y., 2016. Greenhouse gas emissions and reactive nitrogen releases during the life-cycles of staple food production in China and their mitigation potential. Sci. Total Environ. 556, 116.
Xing, G.X., Zhu, Z.L., 2002. Regional nitrogen budgets for China and its major watersheds. Biogeochemistry 57, 405–427.
Yan, Z.J., Chen, S., Li, J.L., Alva, A., Chen, Q., 2016. Manure and nitrogen application enhances soil phosphorus mobility in calcareous soil in greenhouses. J. Environ. Manag. 181, 26–35.
Yu, W.S., 2016. Agricultural and agri-environment policy and sustainable agricultural development in China. Popul. Rep. 247, 1–46.
Zhang, F., Cui, Z., Chen, X., Ju, X., Shen, J., Chen, Q., Liu, X., Zhang, W., Mi, G., Fan, M., 2012. Integrated nutrient management for food security and environmental quality in China. Adv. Agron. 116, 1–40.
Zhang, J.J., Guo, C.X., Zhang, Y.G., Han, P.Y., Zhang, Q., 2016a. Spatial characteristics of nitrogen flows in the crop and livestock production system of Shanxi Province, China. Acta Ecol. Sin. 36, 99–107.
Zhang, J.Y., Lv, C., Tong, J., Liu, J.W., Yu, D.W., Wang, Y.W., Chen, M.X., Wei, Y.S., 2016b. Optimization and microbial community analysis of anaerobic co-digestion of food waste and sewage sludge based on microwave pretreatment. Bioresour. Technol. 200, 253–261.
Zhang, W.F., Dou, Z.X., He, P., Ju, X.T., Powlson, D., Chadwick, D., Norse, D., Lu, Y.L., Zhang, Y., Wu, L., Chen, X.P., Cassman, K.G., Zhang, F.S., 2013. New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. Proc. Natl. Acad. Sci. 110 (21), 8375–8380.
Zhang, X., Xu, X., Liu, Y., et al., 2016c. Global warming potential and greenhouse gas intensity in rice agriculture driven by high yields and nitrogen use efficiency. Biogeosciences 13 (9), 2701–2714.
Zhou, J.B., Jiang, M.M., Chen, G.Q., 2007. Estimation of methane and nitrous oxide emission from livestock and poultry in China during 1949-2003. Energy Policy 35, 3759–3767.
Zhu, Z.L., Chen, D.L., 2002. Nitrogen fertilizer use in China-contributions to food production, impacts on the environment and best management strategies. Nutrient Cycl. Agroecosyst. 63, 117–127.